

Nitrogen Dynamics and Nitrate Leaching in Intensive Vegetable Rotations in Highlands of Central Java, Indonesia

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Received 28 March 2016/ accepted 30 April 2016

ABSTRACT

High rainfall intensity is major factor governing leaching process, where leaching is often the most important process of N loss from the field and lead to agricultural environmental pollution. In order to measure the movement of mineral-N in soil profile, a field research had been conducted in two sites of center vegetable farming area with six farmer cooperators in Central Java, Indonesia. Regular soil sampling was done from Improve Practice (IP) and Farmer Practice (FP) treatment for three planting seasons during 2007. Almost all treatments FP applied higher rate of N fertilizer compare to IP, but it was not reflected in N profile. Comparison of predicted and measured mineral N content was simulated using Burns α model, then the closeness of the estimation and measured calculated using Coefficient of Residual Mass (CRM) calculation as an indicator with 0 as ideal value. Out of 9 measurements of IP and FP treatment, eight and seven measurements had negative CRM representing a slight overestimation. The $\text{NO}_3\text{-N}$ loss estimated using the Burns α model for IP and FP was in average of 67% for IP and 71% for FP of total N fertilizer added or 67% for IP and 76% for FP of total-N surplus, respectively. The calculation of potential nitrate concentration (PNC) at 1 m soil depth at the end of the third season showed a high concentration with significant different of IP and FP having mean value of 59.8 and 82.5 mg N L⁻¹. From the gathered data it was obvious that over N fertilization had negative effect to agricultural environment.

Keywords: N loss, improve practice, farmer practice, over N fertilization

ABSTRAK

Intensitas curah hujan yang tinggi adalah faktor utama yang mendorong terjadinya proses pencucian, dimana pencucian ini adalah proses yang sangat penting terhadap kehilangan N dari lahan dan memicu terjadinya polusi lingkungan pertanian. Dalam upaya untuk mengukur pergerakan mineral N pada profil tanah, penelitian lapang telah dilaksanakan di dua lokasi sentra budidaya sayuran dengan enam petani kooperator di Jawa Tengah, Indonesia. Secara periodik sampling tanah telah dilakukan pada petak perlakuan *Improve Practice* (IP) dan *Farmer Practice* (FP) selama tiga musim tanam pada tahun 2007. Hampir semua perlakuan FP mengaplikasikan pemupukan N yang lebih tinggi dibandingkan dengan IP, tetapi tidak terefleksi pada profil N tanah. Perbandingan dari perkiraan dan pengukuran kadar mineral N telah disimulasi menggunakan model Burns α , kemudian kedekatan dari estimasi dan pengukuran dihitung menggunakan *Coefficient of Residual Mass* (CRM) sebagai indikator dengan nilai 0 sebagai nilai ideal. Dari 9 kali pengukuran perlakuan IP dan FP, delapan dan tujuh pengukuran mempunyai nilai CRM negative yang menunjukkan sedikit over estimasi. Kehilangan $\text{NO}_3\text{-N}$ yang diestimasi menggunakan model Burns α untuk perlakuan IP dan FP terdapat pada rata-rata 67% untuk IP dan 71% untuk FP dari total pupuk N yang ditambahkan atau 67% dari perlakuan IP dan 76% perlakuan FP berdasarkan total surplus N secara berturut-tan. Perhitungan dari potential nitrate concentration (PNC) pada 1 m kedalaman tanah pada akhir musim ketiga menunjukkan kadar yang tinggi dengan perbedaan yang nyata untuk perlakuan IP dan FP dengan nilai rata-rata 59,8 dan 82,5 mg N L⁻¹. Dari data yang terhimpun ini sangat jelas bahwa pemupukan N yang berlebihan mempunyai efek negative untuk lingkungan pertanian.

Kata kunci: Kehilangan N, Improve Practice, Farmer Practice, pemupukan N yang berlebih

INTRODUCTION

N is a major plant nutrient required for high yields of agricultural crops. Nitrogen (N) is the nutrient that is most susceptible to transformations affecting plant availability. These transformations include mineralization, immobilization, nitrification, and denitrification, as well as leaching and NH_3 volatilization (Petersen *et al.* 1998). The N in NO_3^- is soluble and mobile and susceptible to transport into groundwater, which has become increasingly degraded (Strebel *et al.* 1989; Spalding and Exner, 1993). Sources of NO_3^- in agricultural soils are inorganic-N fertilizer, manure, crop residue, and soil organic matter. Agriculture is considered to be the primary contributor to NO_3^- contamination of groundwater (Strebel *et al.* 1989; Fraters *et al.* 1998).

Nitrate leaching and contamination of ground water and surface water has become a major environmental problem in Europe and the USA and also increasingly in Asian countries, including China and Indonesia due to over-application of N fertilizers and farmyard manure in rainfed areas (Kurokura *et al.* 2001; Di and Cameron 2002; Zhu and Chen 2002; Arsanti 2008). The threshold recommended by the World Health Organization for drinking water is $50 \text{ mg NO}_3\text{L}^{-1}$, similar to a maximum admissible concentration defined by European Community (EC) legislation and the US Environmental Protection Agency (10 mg N L^{-1}). Rass *et al.* (1999) stated that NO_3^- contamination of groundwater is closely related to corresponding agricultural management practices.

Nitrogen over-fertilization, particularly in vegetable production, occurs in many countries, including Indonesia. Excessive N fertilizer application is therefore very common, while N use efficiency is often low. Data gathered from farmers' practices (Widowati *et al.* 2012), revealed an average ANUE value 16% for two consecutive years; therefore, more than 80% of N added is susceptible to loss. Application of N fertilizer in excess of crop requirements can result in the accumulation of NO_3^- N in the soil profile (Gillard *et al.* 1995; Malhi *et al.* 2001). Although organic material is considered as a slow-release fertilizer, excessive application may cause unintended NO_3^- leaching (Gerke *et al.* 1999) with the percolating water. Leaching is often the most important means of nitrogen loss from field soils (Aulakh *et al.* 2000; Chowdary *et al.* 2004). The movement of a solute is strongly interlinked with the movement of water. Nitrate movement depends on water movement

below the rooting depth of the crops. A greater water supply than the crop requirement (by rain and/or irrigation) is a very simple indicator of the leaching potential (Song *et al.* 2009; Sipahutar *et al.* 2013).

Nitrate leaching in the Andisols of intensive vegetable production regions in Indonesia is hypothesized to be the primary N loss process, as they have low bulk density, are highly permeable and because the climate is monsoonal with high rainfall intensity ($>2,500 \text{ mL yr}^{-1}$). However, no information is available to date on the impact of high organic and inorganic N fertilizer application on N leaching in intensive vegetable cropping systems in Indonesian (*i.e.*, tropical) Andisols.

The objectives of this study were therefore to monitor NO_3^- N profiles in intensively managed Andisols because leaching cannot be directly derived from such mineral N profiles. The second objective was to quantify NO_3^- leaching losses from model calculations using a simple yet robust leaching model.

MATERIALS AND METHODS

Two Andisols located in Wonosobo and Kopeng were used for the measurement of N movement through the soil profile. Samples for initial soil characteristics were taken the day after harvest of the third planting season of the year 2006.

The soils under study have low clay contents. They contain allophane as an indicator of low bulk density soils, and the BD values were between 0.65 and 0.9 g cm^{-3} in the 25 cm upper layer (Table 1). Another physical characteristic of these soils is a high permeability index (from medium to fast), dominated by very fast pore drainage.

Soil Sampling for Physical Properties and Mineral N Profiles

Soil was collected at both locations from Improved Practice (IP) and Farmer Practice (FP) plots, established at each of three farmers' fields, with each plot consisting of 10 individual beds. During 2007, the beds were not covered; however, in 2008, they were covered with plastic mulch to protect the beds from erosion.

The physical characterization of soils was performed at each farmer's field before the start of the experiment, in layers of 25 cm to a depth of 1 m (Table 1).

The soil sampling for mineral nitrogen profiles was conducted during the year 2007 with sampling intervals of 3 weeks. Each soil sample was collected using an auger at four soil depths: 0-25, 25-50, 50-

Table 1. Some soil physical properties and input data for the Burns model.

Site	Soil layer	Bulk Density	Soil texture	Permeability	Field capacity	Wilting point	Initial moisture content
	(cm)	(g cm ⁻³)		(cm hr ⁻¹)	(%)	% vol.	(%)
Wonosobo							
Sudarto	0 - 25	0.65	Loam	11.62	25.7	17.2	32.5
	25 - 50	0.83	Loam	10.96	30.8	19.2	43.1
	50 - 75	0.66	Silt Loam	5.64	42.4	18.7	32.0
	75 - 100	0.60	Silt Loam	3.83	48.6	23.1	55.1
Nurhakim	0 - 25	0.90	Loam	4.98	37.5	22.5	39.1
	25 - 50	0.83	Loam	2.39	49.5	27.1	35.1
	50 - 75	0.75	Loam	11.84	47.3	13.2	28.0
	75 - 100	0.91	Loam	3.92	48.2	24.8	42.9
Cipto	0 - 25	0.81	Loam	15.68	37.1	20.2	29.6
	25 - 50	0.76	Loam	18.04	45.3	31.8	32.1
	50 - 75	0.63	Loam	8.95	46.6	20.5	36.1
	75 - 100	0.74	Loam	27.12	50.5	32.3	40.1
Kopeng							
Nano	0 - 25	0.78	Loam	6.78	23.9	15.8	29.4
	25 - 50	0.66	Silt Loam	7.63	58.1	16.6	55.2
	50 - 75	0.43	Silt Loam	16.44	54.6	17.0	59.1
	75 - 100	0.53	Silt Loam	2.01	55.5	22.3	59.2
Lukas	0 - 25	0.85	Silt Loam	8.13	28.6	18.8	33.4
	25 - 50	0.81	Silt Loam	11.03	31.8	20.9	37.1
			Sandy				
	50 - 75	0.54	Loam	10.33	41.6	15.6	55.3
Ngatemin	75 - 100	0.47	Silt Loam	3.67	58.6	14.9	54.0
	0 - 25	0.82	Loam	6.35	34.3	18.5	44.0
	25 - 50	0.69	Silt Loam	15.63	35.2	25.6	60.0
	50 - 75	0.55	Silt Loam	13.67	36.6	23.9	63.7
	75 - 100	0.49	Silt Loam	11.68	59.8	22.8	48.1

Table 2. Input data set for Burns α .

Required data		
Soil	Source of N	Meteorological data
- Bulk density (mg m ⁻³)	- Potential Nmin (mg kg ⁻¹ d ⁻¹)	- Precipitation (mm d ⁻¹)
- Moisture content at (%)	- N mineralization form organic matter added (kg N ha ⁻¹)	- Temperature (°C)
- initial conditions	- N from inorganic fertilizer added (kg N ha ⁻¹)	- Evapotranspiration (mm d ⁻¹)
- saturation,		
- field capacity	- plant N uptake (kg N ha ⁻¹)	
- wilting point	- Initial NO ₃ content (kg N ha ⁻¹)	
	- Denitrification constant	

75, and 75-100 cm. Maximum care was taken to minimize the contamination of samples by the soil material of the overlying layers. The sampling points were carefully selected to avoid contamination with recently applied mineral or organic fertilizer. The soil samples were then analyzed for NH₄-N+NO₃⁻ N. Ammonium nitrogen was extracted with 1 N KCl

followed by colorimetric analysis with a spectrophotometer at wavelength 636 nm, and NO₃⁻ N was extracted with 0.01 M CaCl₂ followed by colorimetric analysis with a UV spectrophotometer at wavelengths of 210 and 270 nm (Hitachi U-2010, Japan).

To estimate the movement of NO_3^- , we selected the Burns α model for its simplicity and versatility. The Burns α leaching model is an adaptation of the Burns leaching model (Burns 1974) and was chosen because it requires only readily available soil and meteorological data (De Neve and Hofman, 1998; Moreels *et al.* 2003; Chaves *et al.* 2006) as presented in Table 2. Of the numerous leaching models published, this model is one of the few that has been applied to actual field conditions (Scotter *et al.* 1993). Moreels *et al.* (2003) successfully used the Burns α model to predict moisture and nitrate contents in bare fallow soils, and Chaves *et al.* (2006) used this model to predict N release from N-rich crop residues and organic wastes.

One major drawback of the Burns model is that no water content above field capacity can be simulated, thus limiting its use to light textured soils (Moreels *et al.* 2003). To adapt this model to other soil textures, the model was adjusted by adding the α parameter, which denotes the proportion of water above field capacity that drains to the underlying layer (a varies between 0 and 1), which must be specified for each soil layer. This adjustment allows simulations of moisture contents between field capacity and saturation. The value of α for each layer is obtained by calibrating the model using measured soil moisture contents. In this research, we used data from a bromide leaching experiment on these soils for the calibration of α (data not shown). The calibration based on Br⁻ leaching yielded an α value of 1 for all layers and all plots, thus indicating very rapid leaching.

De Neve and Hofman (1998) extended the model with a mineralization module, which calculates and takes into account the N mineralization from soil organic matter and from added organic materials. Moreels *et al.* (2003) further adapted the model to also calculate N losses by denitrification. Because the model was developed for simulating leaching only in bare fallow soils, we need to correct the calculations for N uptake by the crop as follows: each day, the NO_3^- -N concentration in the top layer was calculated, taking into account inputs by mineralization and fertilizer and outputs by leaching from the previous day. This daily NO_3^- -N concentration was then diminished by the daily N uptake by the crop (calculated as total N uptake divided by the length of the cropping cycle in days; *i.e.*, assuming a linear crop N uptake pattern) before calculations for that day were continued. The data on N mineralization from soil organic matter and organic materials were taken from Widowati *et al.* 2012, the total crop N uptake was taken from

Widowati *et al.* 2011, and the denitrification rates were obtained from Anggria (2007). Data used in the running model were from the first and third planting seasons of the year 2007; *i.e.*, only during the rainy season, as there was no leaching in the second planting season.

Meteorological Data

Meteorological data for 2007-2008, namely relative humidity, temperature, evapotranspiration, and cloudiness, were obtained from the closest climate station, while precipitation (which is much more spatially and temporally variable) was collected by the farmer co-operators directly. Data for the Kopeng site were obtained from Meteorological and Geophysical Agency Area II, which is located at Ungaran Sub-district (± 35 km distance), and climate data for the Kejajar-Wonosobo site were obtained from the Tambi Tea Plantation Climate Station, which is located less than 3 km from the Wonosobo sites.

Estimation of Nitrate Concentrations in Soil Water at 1 m Depth

A rough estimation of the nitrate concentration was obtained using the total N and water balance equation according to OECD (1999), as in Maeda *et al.* (2003). The equation for predicting the NO_3^- -N concentration in soil water at 1-m depth is:

$$PNC = \frac{PNP}{EW} \times 100$$

where PNC (mg L^{-1}) is the potential nitrate concentration; PNP ($\text{kg N ha}^{-1} \text{ yr}^{-1}$) is the potential nitrate nitrogen present in soil, which in this calculation represents the total N-balance within one year; and EW (mm yr^{-1}) is the excess water (the difference between water input and evapotranspiration). Evapotranspiration was calculated using the Penmann equation. Maeda *et al.* (2003) obtained PNP by subtracting the amount of N uptake, which refers to N in all parts of the crops removed from the plots from that of the total N application in a year. However, the authors assumed all N from the applied organic matter in their study to be available within one observation, which obviously is an oversimplification. Here, the total N-balance for the calculation of the PNP was obtained directly from the N-balance results (Table 3). Based on these N-balance calculations, the data used here to calculate the PNP more realistically represent the potential nitrate present in the soil, as

Table 3. Nitrate loss estimation based on Burns_α model simulation and relative to N-total added and N-surplus.

Farmer	Season		N-total added	N-loss	Percentage	
					N-total added	N-surplus
Name			kg N ha ⁻¹			
Kopeng						
Nano	III	IP*	448	162	36	52
		FP*	401	171	43	52
Lukas	I	IP	nd**	nd	nd	nd
		FP	nd	nd	nd	nd
	III	IP	344	117	34	46
		FP	370	211	57	75
Ngatemin	I	IP	nd	nd	nd	nd
		FP	nd	nd	nd	nd
	III	IP	448	283	63	117
		FP	745	421	56	80
Wonosobo						
Nurhakim	I	IP	397	435	110	99
		FP	410	417	102	88
	III	IP	397	56	14	11
		FP	430	48	11	11
Darto	I	IP	282	69	25	22
		FP	902	84	9	10
	III	IP	397	442	111	110
		FP	430	787	183	190
Sucipto	I	IP	397	301	76	62
		FP	621	232	37	35
	III	IP	297	406	137	88
		FP	696	984	141	144

* IP = improve practice; FP = farmer practice

** nd = no adequate data

the parameters of N-balance were calculated in a more comprehensive way than in Maeda *et al.* (2003).

Statistical Analysis

Significant differences in those parameters between IP and FP were calculated by t-tests using SPSS v15.0. To evaluate model performance, an analysis of coefficient of residual mass (CRM) and the modeling efficiency (EF) were conducted. The CRM and EF were calculated as (Vereecken *et al.* 1991; Moreels *et al.* 2003):

$$CRM = 1 - \frac{\sum_{j=1}^N P_j}{\sum_{j=1}^N O_j}$$

$$EF = 1 - \frac{\sum_{j=1}^N (P_j - O_j)^2}{\sum_{j=1}^N (O_j - \bar{O})^2}$$

Where P_j are the simulated values, O_j are the measured values, \bar{O} is the average of the observed values and N is the number of data pairs. The EF provides a comparison of the efficiency of the model to the efficiency of describing the data as the mean of the observation. The optimum value for EF is one, while the CRM should be as close to zero as possible. The EF can become negative, indicating that the observed mean is a better estimate of the observations than the model predictions. The coefficient of residual mass CRM indicates whether the observed data are overestimated (CRM<0) or underestimated (CRM>0) by the simulations (bias).

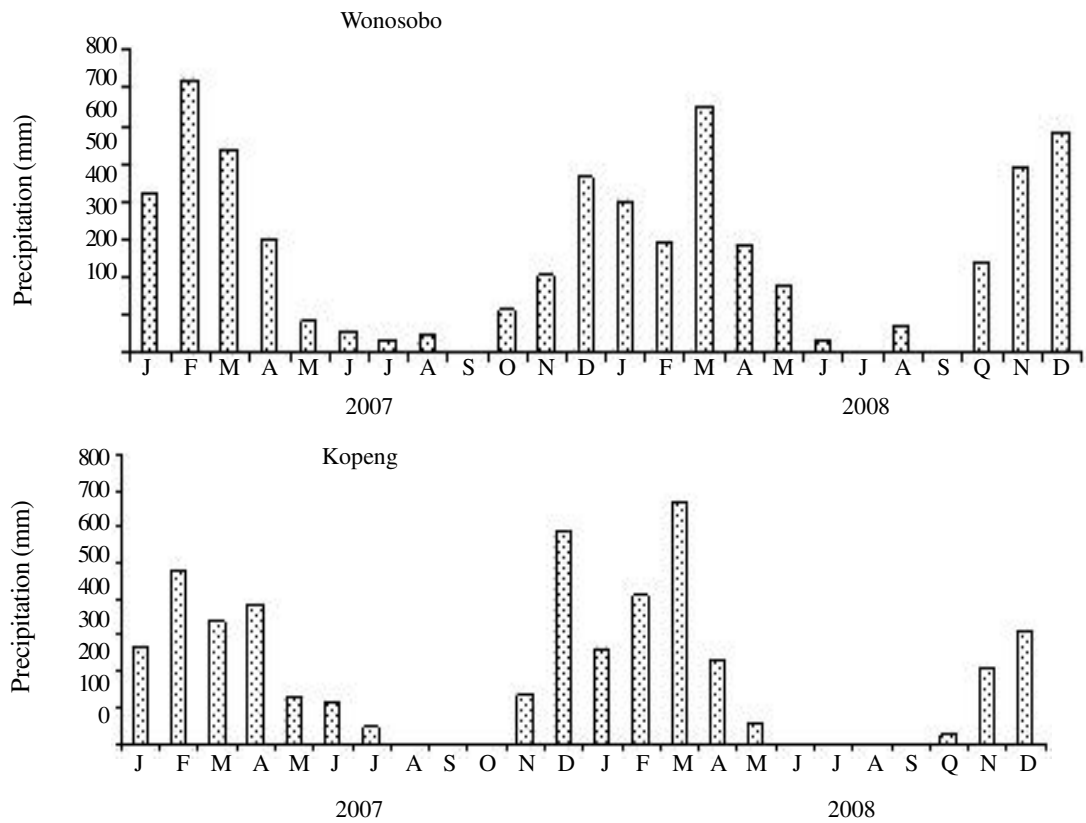


Figure 1. Monthly precipitation (mm) at Wonosobo and Kopeng sites year 2007-2008

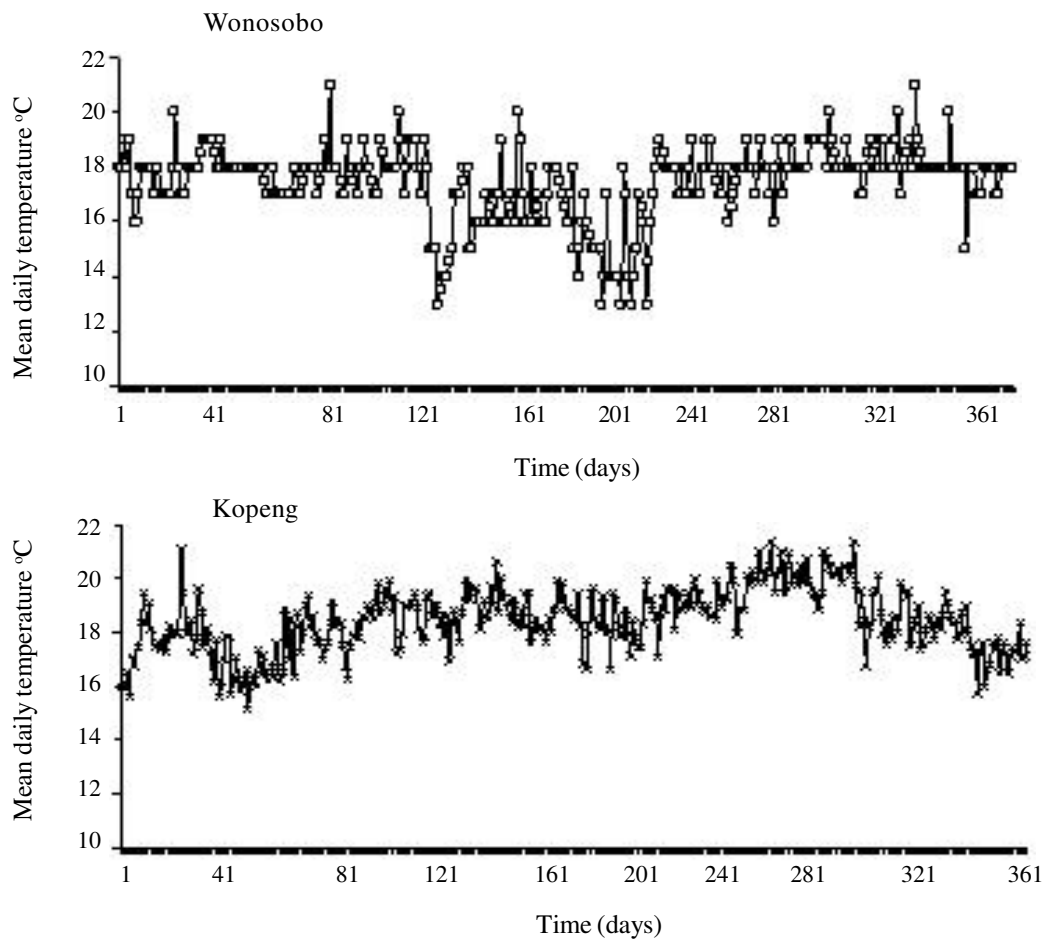


Figure 2. Mean daily temperature of Wonosobo and Kopeng sites year 2007-2008.

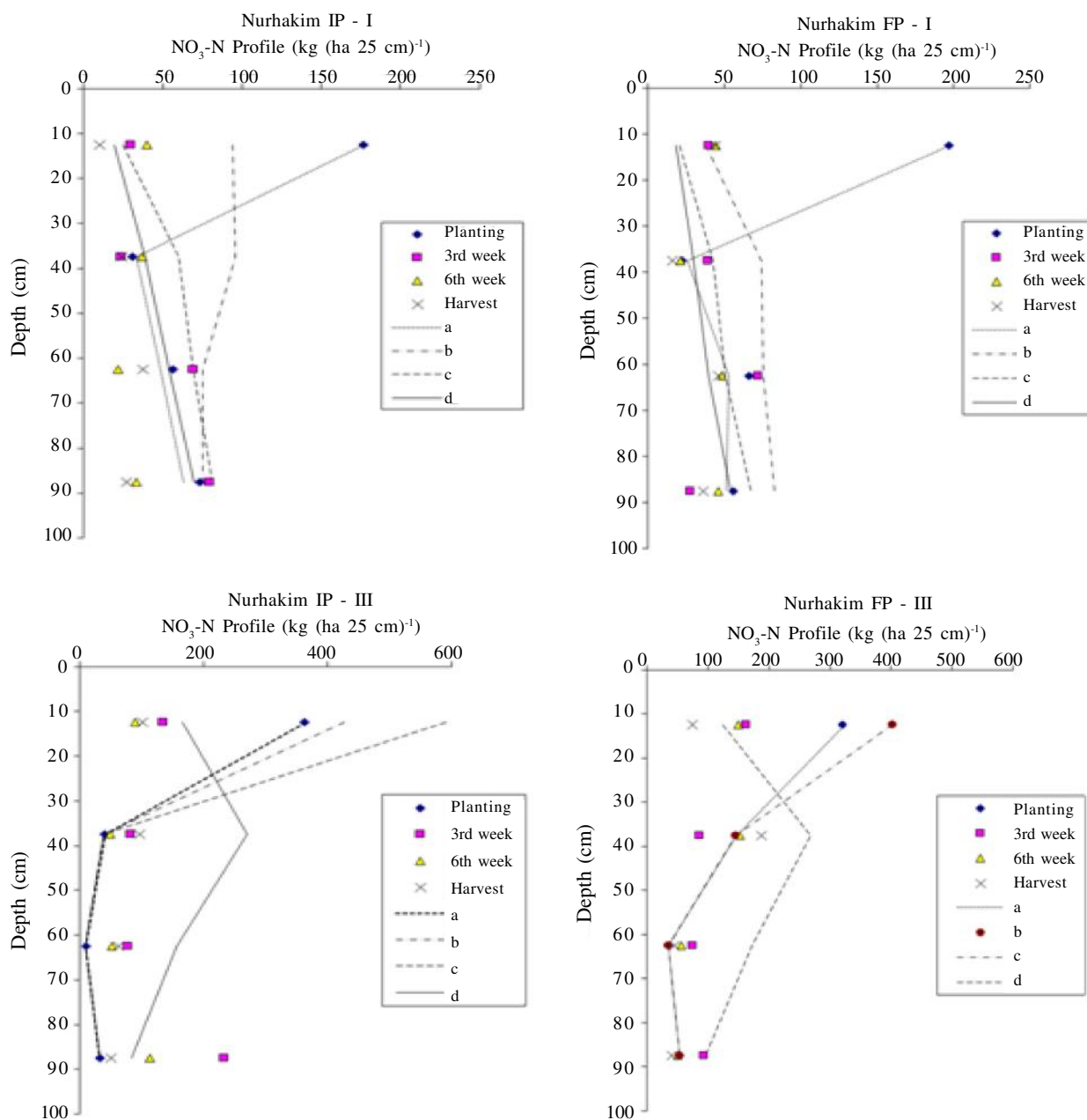


Figure 3. Representative soil mineral N-profiles from the Wonosobo site. Measured (symbols) and simulated (lines a-d) distribution of mineral N over the different soil layers at four sampling times on Nurhakim’s site – Wonosobo year 2007.

RESULTS AND DISCUSSION

Rainfall and Temperature Pattern

The Wonosobo site received more rain on average (3034 mm year⁻¹) than the Kopeng site (2234 mm year⁻¹), and the rain was concentrated in November to May (Figure 1). There are significant differences concerning the rainfall intensity during the wet and dry seasons. During the dry season (June to October), some farmers decided to leave the field bare, primarily where irrigation water was not available.

The average temperature at the Wonosobo experimental site ranged from 13 to 21 °C, with an average of 17.5±1.4 °C; at the Kopeng site, the temperature ranged from 15 to 21 °C with an average of 18.4±1.2 °C (Figure 2). These temperatures are appropriate for optimum growth of vegetables adapted to the highland climate.

Soil Mineral Nitrogen Profile

The soil mineral N profile (NO₃-N) in these Andisols was highly dynamic, both between locations and temporally (Figure 3 and 4). Almost all FP treatments applied much higher rates of N fertilizer

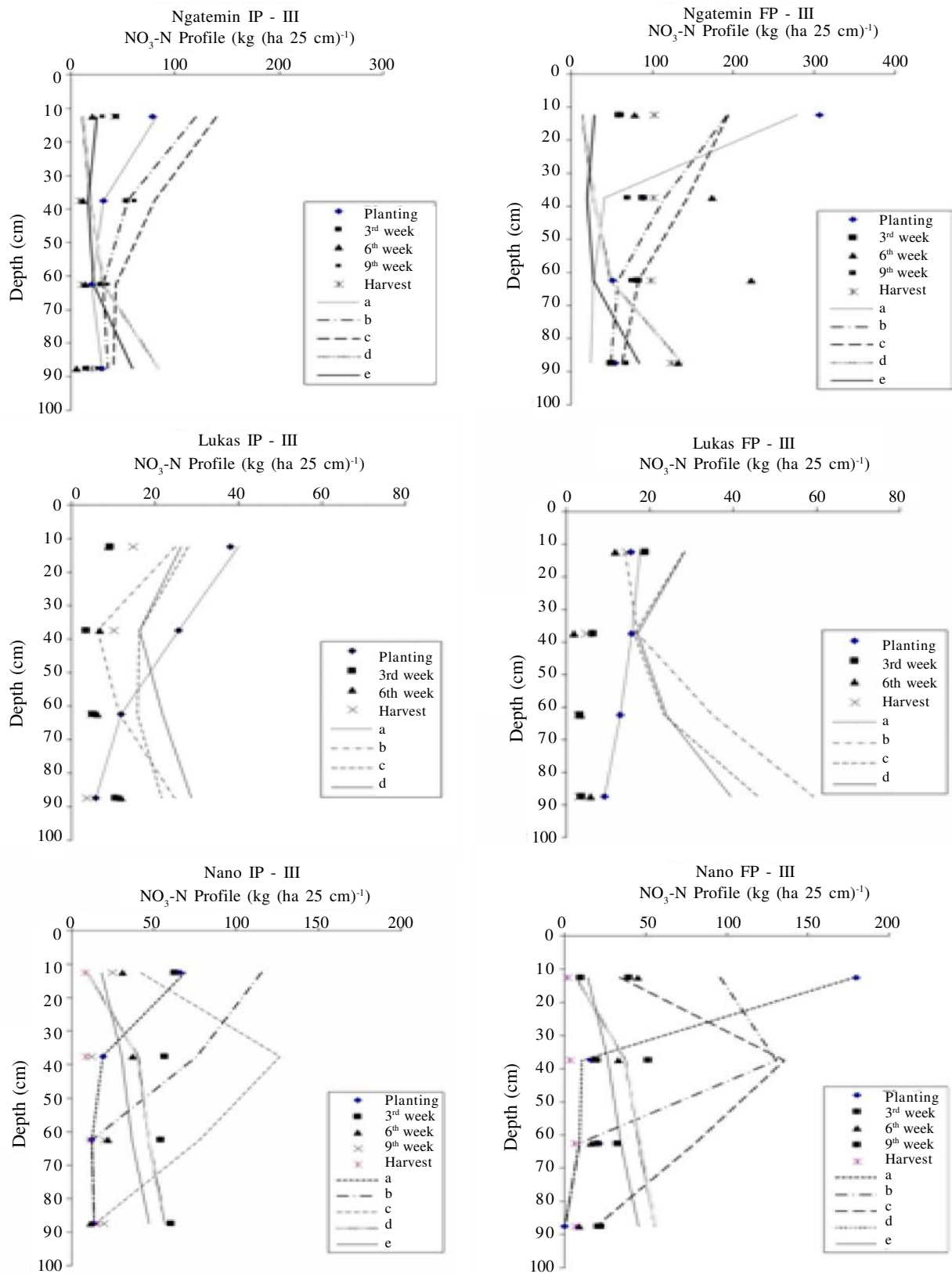


Figure 4. Representative soil mineral N-profiles from the Kopeng site. Measured (symbols) and simulated (lines a-d) distribution of mineral N over the different soil layers Kopeng year 2007.

compared to the IP treatments (Widowati *et al.* 2011), but this was surprisingly not clearly reflected in the total mineral N in the soil profiles. Of the 36 growing seasons, only five had significantly greater

total mineral N in FP, whereas in the rest of the plots, only a tendency towards higher mineral N was observed. During the third week after planting, the mineral N of the plots tended to increase. The

second fertilization was applied during the fourth week after planting but was not followed by an increase in the mineral N content in the second observation (42 days).

Levels of NO₃ measured in the soil monitored at a depth of 0-25 cm have been found to change very rapidly. Several researchers report an increased concentration of residual NO₃-N in soil profiles after the application of large amounts of N to different cropping systems (Malhi *et al.* 2001; Gillard *et al.* 1995) and N mineralization from soil organic matter and organic fertilizer, while decreased nitrate content occurs due to plant uptake, immobilization, denitrification, and leaching.

Mineral nitrogen in the soil is easily transformed and translocated, which is reflected in the N-profile measurements over time. Although a portion of the N (NH₄⁻-N) from fertilizer and organic matter is adsorbed by soil particles and taken up by the plants, the remaining N was estimated to be highly concentrated in the form of NO₃ and susceptible to leaching into a deeper layer. A simulation of the estimation of factors affecting nitrate leaching was conducted by LilBurnse *et al.* (2003); they reported that soil type, climate, and sowing date explained approximately equal amounts of the variance in nitrate leaching, whereas fertilizer application explained only approximately one-third of the variance of the other inputs.

The results of the mineral N dynamics between IP and FP indicated that there was no specific pattern. Although FP typically applied higher total-N, a greater accumulation of mineral nitrogen did not always occur. The reason for this is not very clear. One reason may be that residual N from previously applied manure was mineralized and may have been available for leaching in the continuously managed fields (Angle *et al.* 1993, Thomsen *et al.* 1993; Bergstrom and Kirchmann, 1999). Another reason may be the very specific nature of these soils, as discussed below.

Listen Read Phonetically Simulation of Nitrate Movement and Loss Using the Burns α Model

Nitrate loss simulated from IP and FP averaged 252.3 kg NO₃-N ha⁻¹ (ranged from 56 to 442 kg NO₃-N ha⁻¹) and 372.7 kg NO₃-N ha⁻¹ (ranged from 48 to 984 kg NO₃-N ha⁻¹), respectively, but were not significantly different (Table 3). Comparing the simulated N losses to the total N added generated a percent loss varying from 14 to 137% (average 67%) for IP and 9% to 183% (average 71%) for FP. The calculation of N-balances (Widowati *et al.* 2011) revealed large N surpluses, and the relative percentage of simulated NO₃ loss to N-surplus produced varied from 11 to 117% (average 67%) for IP and 11 to 141% (average 76%) for FP.

The simulation results of N leaching using the Burns_α model obtained eight and seven negative CRM values over nine seasons for IP and FP, respectively. The CRM value of IP ranged from -1.869 to 0.165, and the value of FP ranged from -2.030 to 0.165. The EF (which is ideally equal to 1) was low in most of the cases, and the largest EF value reached 0.725.

Nitrate Leaching Simulation Using the Burns α Model

Direct measurements of NO₃⁻ leaching in soil are very difficult. The measurement of mineral N profiles over time only gives an approximate idea of NO₃⁻ leaching losses. The use of porous suction cups allows to measure the NO₃ concentration in the water moving through the soil. However, the use of porous cups also only provides an approximate idea about NO₃⁻ leaching, as measured concentrations need to be transformed into fluxes. Moreover, measurements of NO₃⁻ concentrations with porous cups are often problematic, and e.g. much care has to be taken for installation. The only direct way of

Table 4. Potential nitrate concentration at 1 m soil depth.

Site/Farmer name	N-balance		PNC	
	IP	FP	IP	FP
Wonosobo	kg N ha ⁻¹		mg N L ⁻¹	
Nurhakim	1505	1807	76.9	92.3
Sudarto	1128	1866	57.6	95.3
Sucipto	1351	1869	69.0	95.5
Kopeng				
Nano	811	901	52.6	58.4
Lukas	795	1104	51.5	71.6
Ngatemin	790	1267	51.2	82.1

measuring NO_3^- leaching is by the use of weighing lysimeters. However, such instruments are very expensive and only available at a very limited number of locations.

As the leaching of soil NO_3^- -N is very difficult to measure directly in situ, a simulation model like the Burns_α model has to be used to predict nutrient losses to the environment (Moreels *et al.* 2003). Efforts to simulate the movement of NO_3^- -N using the Burns_α model were not very successful for the soils in this study, as reflected by the mostly overestimated results (negative CRM) and often poor EF values. Given the very low N use efficiency in these rotations, we would expect most of the surplus to be lost through leaching over time (1-2 years). However, the total N loss by leaching was much less than the N surplus in most cases, and was always less than the total N added. Because of the relatively large mineralization rates, we would expect most of the N added to be lost by leaching. This is in contrast with the good results obtained with the model in previous studies (*e.g.* De Neve and Hofman 1998; Moreels *et al.* 2003; Chaves *et al.* 2006), and here we discuss a number of reasons for the limited success of the simulations.

A first reason for the relatively poor simulation results could be the nature of the soils in this study. To our knowledge, this simulation model has not yet been applied in Andisols, and in general there have been very little studies on NO_3^- leaching in Andisols using simulation models. The nature of these soils (high permeability) combined with the very intensive rainfall during the rainy season result in very fast leaching. Possibly the model was not able to capture this fast leaching rate. The calibration of the model (using leaching experiments with Br^- as a tracer) gave an α value of 1, which is the maximum. This value corresponds to immediate leaching of all water above field capacity to the underlying layer within one-time step. Perhaps the time step used normally with this model (1 day) was too coarse to capture the very fast dynamics of leaching. Another reason related to the nature of the soils could be the difficulty in measuring accurate values of field capacity. Because of the very low bulk density, physical soil properties have rather high uncertainties. If the real field capacity of these soils is lower than measured here, than the actual leaching would be faster than the leaching simulated.

Another reason for the lack of agreement between measured and simulated NO_3^- profiles may be the way the N dynamics in soil were implemented in the model. The original Burns_α model did not include a plant N uptake module. The crop N uptake

was implemented in the model in a simplified way, namely dividing the total crop N uptake over the entire growing season over the number of days, and this amount was subtracted each day from the amount of NO_3^- -N present in the soil profile. We were also unable to take into account the rooting depth of the different crops. However, the error introduced by an inaccurate simulation of the crop N uptake would have a relatively small impact on the simulation of NO_3^- leaching, because crop N uptake often was only a minor fraction in the total N balance of these rotations.

Finally, anion exchange of NO_3^- could also influence simulation results, because Andisols may have significant anion exchange capacity. However, lab measurements indicated that the anion exchange of NO_3^- was negligible in these soils (data not shown).

Estimation of NO_3^- Concentration in the Soil Water at 1 m Depth

The average total evaporation was 1033 mm year⁻¹ and 942 mm year⁻¹ for Wonosobo and Kopeng, respectively. The average excess water (EW) was 2073 mm year⁻¹ for the Wonosobo sites and 1275 mm year⁻¹ for the Kopeng sites, respectively.

The IP and FP nitrogen balance results were highly positive, as the nitrogen fertilizer was heavily applied (Widowati *et al.* 2012). The Potential Nitrate Concentration (PNC) calculation results for IP and FP therefore demonstrated high concentrations, with averages of 59.8 mg N L⁻¹ and 82.5 mg N L⁻¹ in 2007, respectively (Table 4). The average PNC values of IP and FP were significantly different ($P < 0.05$).

Potential Nitrate Concentration

To estimate the movement of nitrate, nitrogen surplus accumulated as nitrate-nitrogen can be calculated in the soil at 1 m soil depth (PNC) (Maeda *et al.* 2003). Agricultural activity using organic and inorganic N fertilizers affects this PNC value. Improved practice tended to decrease the PNC value compared to farmer practice. However, IP still had a high PNC value compared to the maximum admissible concentration of 11.3 mg NO_3^- -N l⁻¹ (the 1991 Directive 91/676 – European Community (EC) legislation), which was extended to apply to surface freshwater and groundwater intended for abstraction for drinking water and to freshwater, estuaries and coastal waters liable to eutrophication (Rodda *et al.* 1995). This is an indication that the vegetable farming systems in both locations experience over-fertilization; therefore, inefficiency occurred and

contributed to very high soil pollution. Maeda *et al.* (2003) also reported that excessive N from chemical fertilizer applied to Andisols can cause NO₃⁻ leaching at 1-m depth under the Japanese climate (Asian monsoon).

CONCLUSIONS

Nitrate leaching is a problem that often occurs in intensive vegetable production centers in Indonesia. Measurements must be taken to monitor and provide evidence to farmers to improve the understanding of this problem. Although the FP always applied more nitrogen, a greater mineral N content was not always observed in the soil profile. Simulation results obtained from the Burns_a model on Andisols in general did not agree very well with the measured nitrate profiles, resulting mostly in overestimations of the actual values. The likely factors responsible for these overestimations were the very particular nature of these volcanic soils (low BD, uncertainties about soil physical properties), exceptionally high rainfall intensity and the way in which the crop N uptake was taken into account. Nitrate leaching can also be approximated by measuring the potential nitrate concentration (PNC) at 1 m soil depth. PNC values calculated here were very large. Although only approximate, the amounts of nitrate leaching and high PNC values found here are quite alarming, and farmers and policy makers should be made more aware of this issue that is threatening water quality and is a sign of highly inefficient N use.

ACKNOWLEDGEMENTS

This research was in the framework of the VLIR-OI project "Nitrogen balances in vegetables production in Central-Java: a tool for improving N use efficiency for smallholder farmers (ZEIN2005PR307). We acknowledge the financial support from the Flemish Interuniversity Council (VLIR-UDC). Many thanks go to Diah Setyorini, Sukristyonubowo, A. Kasno, Ibrahim A. Sipahutar, Didik Sukristyonubowo for improvement of the manuscript, laboratory, and field research assistance.

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